# Unifying concepts

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*Chapter 2*

# UNIFYING CONCEPTS

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## 2.1 INTRODUCTION

**Restoration ecology** is an applied natural science that lies at the intersection with the social sciences, but can also help us leap from that broad platform into the realm of transdisciplinary science and problem solving, which we will discuss below. Restoration ecology is thus truly a 'new frontier', as first noted by one of the most notable and prolific pioneers in the field, in the introduction to the book he edited (Cairns 1988), which was one of the very first books to appear on this topic.

In this book, we focus on the ecological foundations of restoration ecology. We feel strongly that restoration efforts must aim to restore entire ecosystems, and not just focus on parts of them, or other derivative goals. Increasingly, we hear and read about the need to 'restore' **biodiversity**, or **ecosystem services**, but these goals are ultimately vain if we do not succeed in restoring living, dynamic ecosystems, and figuring out how to help them be self-sustaining. It is difficult or impossible to 'restore' or rather **reintroduce** species populations in a given site, without 'restoring' the abiotic environment necessary for the persistence and reproduction of those species, including the networks of interactions with many other species that occur in a well-functioning **ecosystem**. Conversely, **biotic communities** strongly influence the abiotic environment, and without a full complement of native species, autogenic or self-sustaining ecosystems – the ultimate goal of ecological restoration – will not be attained (MacMahon & Holl 2001). Thus, we endorse the definition given in the *SER Primer for Ecological Restoration* we cited already, namely, that **ecological restoration** is 'the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed' (SER 2004).

Note the emphasis in that definition on the idea of assisting the recovery of an ecosystem, and not just a species. The definition explicitly assumes that something has been lost, or gone wrong at the level of a system, and, secondly, it implies that we can and should try to understand how ecosystems respond to interventions of all sorts, including efforts to help them recover. Ecological restoration is interventionist and systems-oriented by nature, as opposed to traditional conservation, that was about reducing human pressure or 'keeping our hands off' certain areas of land or wetland set aside for protection of one or an assembly of species. It is hands on, and is, by definition, applied at the level of whole ecosystems.

The corresponding field of science called restoration ecology can take various approaches to the task of providing knowledge that will help put ecosystem recovery in motion. New theories and syntheses, predictive models and the testing of hypotheses through experiments and careful monitoring and evaluation of ongoing projects are the primary means to achieve that end. Additionally, outreach and collaboration with people from other academic disciplines, in both the natural sciences (e.g. conservation biology and landscape ecology) and the social sciences, including economics, as well with nonscientists and professionals, is essential. That will require engaging in the 'entire restoration process' (Cairns & Heckman 1996). In this chapter, then, we focus on the major unifying concepts relevant to both fundamental and applied ecology, but start with the notions of inter- and transdisciplinarity.

## 2.2 INTER- AND TRANSDISCIPLINARITY

Restoration ecology draws knowledge, ideas and data from disciplines as diverse as landscape ecology (including geomorphology and hydrology), community ecology along with soil and water physics, and chemistry at the ecosystem scale, as well as physiology and genetics at the level of organisms and populations. But as mentioned, to address and engage the 'entire restoration process', we must incorporate the socio-economic sciences (e.g. Mascia *et al*. 2003). This implies cross- or *interdisciplinarity*, which is what happens when concepts, models, methods and findings of different scientific disciplines are merged together and integrated to address an idea, or to solve a societal problem (Schoot Uiterkamp & Vlek 2007).

Scientists need to cross traditional lines and work together in the essential arena of environmental amelioration and management. The word 'transversal' – which means cross-cutting – is rarely used in English as an adjective, and yet it beautifully describes what is needed: not just a summing of skills, but also an actual breaking of new ground, thanks to original or 'lateral' thinking, resulting from a new juxtaposition and combination of approaches. In order to help **stakeholders**, and society as a whole, in the urgent task of restoring, repairing and rehabilitating the natural and **socio-ecological ecosystems** on which we all depend, new synergies of this kind are clearly called for. All of the categories of 'nature' referred to in Chapter 1 – 'wilderness' or near natural systems, semi-natural ecosystems and production systems – not to mention the fourth category in Figure 1.3, namely, over-exploited systems – can all be approached from ecological, economic, social, cultural and political perspectives, as per Figure 1.1. Of course, the intensity of human impact in each category may be strong, moderate or close to nil, and of course such a simplistic typology can distract from the tremendous complexity of landscapes (and seascapes) that occur around the world. Yet for now, it can help advance discussion, especially in an interdisciplinary setting.

Making interdisciplinarity 'work' is actually a huge achievement, but **transdisciplinarity** takes us several steps further. We appreciate, by the way, that the term transdisciplinarity can seem daunting, especially when you learn that common synonyms or related terms and expressions are being used in the academic literature such as 'post-normal science', 'mode-2 knowledge', 'integration and implementation sciences' and 'interdisciplinary and interprofessional problem solving' (see e.g. Scholz *et al.* 2000). Following Max-Neef (2005), who provided an approach that we adopt here, transdisciplinarity implies that we must cross not only traditional boundaries between scientific fields and engineering, design or management professions, but also the unofficial but deeply entrenched frontiers between scientific and nonscientific habits, techniques and social communities. Indeed, the goal is to get professionals, scientists, public officials, landowners, businesspeople and leaders of local people – in brief, all relevant stakeholders – involved in the restoration process and the transition to **sustainability**.

We will now illustrate the importance of inter- and transdisciplinarity in ecological restoration with a brief introduction regarding values followed by a presentation of the key notions of **natural capital** – a stock or asset, and **ecosystem services** – the dividends which flow to human society from natural capital. Thereafter, we discuss a core group of fundamental ecological concepts related to three different levels of interest in restoration ecology, respectively focusing on the reintegration of **landscapes**, the repair of degraded or impaired **ecosystems** and finally the salvation and protection of endangered **biodiversity**.

### 2.2.1 Determining values of nature

When it comes to determining how people value 'nature', it is important to recall that 'monetary value' is just one among many options, or proxies in economic terms, that can be applied. But it is an important and convenient one, however, for two reasons. Firstly, money is something that everyone understands, and which therefore facilitates communication. Secondly, money – and the postulates of economists and economic pundits – plays a huge role in our lives as citizens today, whether we approve of this state of affairs or not. The good news from conservationists' perspective is that in recent years there is a new school of thought in economics called **ecological economics** (Daly & Farley 2010). This approach is helping to mainstream nature conservation and restoration objectives into political decision-making and negotiation processes at national and international levels. Costanza and Daly (1992), and many others since, have argued that 'natural capital' has become a limiting factor for both human well-being and economic sustainability (elaborated in Aronson *et al.* 2006, 2007a; Blignaut *et al.* 2007). De Groot *et al.* (2002) developed a framework to help integrate values of **ecosystem goods and services** in the process of decision making.

### 2.2.2 Ecosystem goods and services, natural capital

The *Millennium Ecosystem Assessment* (MA 2005) defined ecosystem goods and services as natural products and processes generated by ecosystems that sustain and fulfil human life (cf. Daily 1997). We prefer, however, to adopt the typology provided by the interdisciplinary, UN- and EU-funded project called *The Economics of Ecosystems and Biodiversity* (TEEB), that distinguishes between **ecosystem processes and functions**, **ecosystem services** and **human well-being** in terms of the benefits and values people perceive or assign to those services (de Groot 2010). This becomes still more clearly interdisciplinary when we see that the term **natural capital** is used as an economic metaphor for **ecosystems** and **biodiversity** (Neßhöver *et al.* 2011). Natural capital is in fact a broad concept indicating the limited stocks of both physical and biological natural resources found on Earth (Aronson *et al.* 2007a).

According to the MA (2005) and TEEB (2010, 2011), there are four, partially overlapping types: (1) renewable natural capital (the restoration of ecosystems), (2) nonrenewable natural capital (subsoil assets, e.g. petroleum, coal and diamonds), (3) replenishable natural capital (e.g. the atmosphere, potable water and fertile soils) and (4) cultivated natural capital (e.g. crops and forest plantations). To help show how this notion provides a clear and usable model of the relationship between ecosystems and society, we quote from Aronson *et al*. (2007b):

> The restoration of natural capital is any activity that integrates investment in and replenishment of natural capital stocks to improve the flow of ecosystem services, which enhances all aspects of human well-being. In common with ecological restoration, natural capital restoration is intended to improve the health, integrity, and self-sustainability of ecosystems for all living organisms. (p. 5)

This statement implies a diversity of rationales, both scientific and nonscientific. It also brings together various kinds of values that are all necessary for an 'entire restoration process' or what Clewell and Aronson (2007) called *holistic restoration*. These include (1) ecological values, based on ecological science and what it can tell us about sustainability at population, community and ecosystem levels, (2) sociocultural values, based on equity, intergenerational justice, and cultural customs and perceptions and (3) economic values, based on efficiency and cost effectiveness combined with the ecological economics caveats that natural capital is the basis of all economies. This approach was adopted in the Millennium Ecosystem Assessment (MA 2005) and further elaborated in the global assessment on the economics of ecosystems and biodiversity (TEEB 2010, 2011).

Weesie and van Andel (2002) modified this model, aiming at explicitly including non-anthropocentric, or biocentric, values in the valuation system. Clewell and Aronson (2007) added personal, psychological and spiritual values, for example those resulting from aesthetic motivations, to this triple valuation, further illustrating the need for transdisciplinarity in the valuation of nature. In the second chapter of the book *Restoring Natural Capital*, Blignaut *et al*. (2007) expressed this same sentiment from a transdisciplinary perspective.

### 2.2.3 Setting goals of restoration

For the full (i.e. democratic and participatory) restoration process, essential for ecological restoration projects to be successful over time, we need transdisciplinarity. Goal setting and the choice of goals are among the most important decisions to be taken at the interface between science and society. This should come after common values have been discussed by stakeholders and project managers, not before! In other words, before starting to actually 'do' ecological restoration, we need to know 'why' we should try to restore damaged or altered ecosystems and to what state or condition or trajectory we wish to see them move (Higgs 1997; Clewell & Aronson 2006, 2007). Ecologist Margaret Palmer and coworkers (Palmer *et al*. 2005) spoke of a 'guiding vision' to help define what we – as a society – want to achieve in a restoration context. This in turn demands, in our view, insight and discussion among stakeholders on the causes of degradation in an historical perspective. It is the task of the science of restoration ecology to search for laws and general rules, and to develop applicable concepts and theories, including, very simply, how to get from State C 'back to' the State A of Figure 1.2.

In Chapter 1, we distinguished three different levels of ambition in the broad field of ecological restoration, as exemplified by the terms restoration, rehabilitation and reclamation. Here we elaborate a bit more on these terms, to illustrate how goal setting depends on collaboration between parties. As noted already, the term **ecological restoration** is often used in a very broad and rather vague way, to mean bringing a site, or place, or ecosystem 'back' to something called or considered as 'original', 'initial' or, more precisely, 'pre-disturbance' conditions in the sense employed in Chapter 1 (Figures 1.2 and 1.3). **Ecological rehabilitation**, in the broad sense, is the improvement of ecosystem functions without necessarily achieving or even seeking a full return to 'pre-disturbance' conditions. A rehabilitated site will be similar in ecological functioning to the reference system (as discussed in this chapter) and contain similar but not necessarily the same organisms. Emphasis is generally on restoring ecosystem processes and functions so as to increase the flow of services and benefits to people. But, nonhuman members of ecosystems are considered as well, and as in restoration, ecological rehabilitation implies consideration of an ecosystem of reference – a concept we will develop further in what follows. Finally, **reclama-**

**tion** is simply about 'improving' a degraded or 'useless' site and making it useful again. The goal is to bring the site to, or back to, a condition considered desirable and sustainable for people, whether the use is for production (e.g. for grazing livestock, for recreation or for something else). In many languages, there is no clear homologue for this word, and terms like 'environmental recuperation or revitalization', 'enhancement' or 'amelioration' are more readily understandable. 'Clean-up' is often a major part of such operations, and new terms exist for this activity as well, such as 'bioremediation' and 'phytoremediation', as applied to oil spills, secondarily salinized areas and other areas that have suffered pollution or massive denaturalization.

It is important to recall that at the scale of whole **landscapes**, where **socio-ecological ecosystems** co-occur and interact with each other, all three activities can be planned and pursued simultaneously. Once the goals for a given landscape unit – restoration, rehabilitation and/or reclamation – have been set, indicator values can be used for diagnostic purposes, as early-warning indicators of deviations, or otherwise in the process of piloting a system along a target **trajectory**. In this book we will not go into monitoring and evaluation of socio-economic, cultural and moral values. Ecological evaluation values are often based on **indicator species**, members of a biotic community that have been shown (by experience or by scientific calibration) to be characteristic of certain environmental conditions and sensitive to changes therein (e.g. Ellenberg *et al*. 1991 for plants; Carignan & Villard 2002 for multiple species; Harris 2003 for microorganisms). They can be used to qualify the direction of changes in an ecosystem, that is, by distinguishing between developing and degrading ecosystems. Note, however, that quantitative indicator values of species that have been asserted in a specific region may not be applicable to, and thus have to be calibrated for their use in, the environmental context of other regions.

### 2.2.4 Reference ecosystems

Central to the goal-setting process in projects and programmes of ecological restoration is the concept of **reference ecosystems**. The choice or construction of a reference ecosystem, or more simply 'the reference', in restoration ecology consists of identifying one or more natural, or seminatural, ecosystems (or descriptions thereof) which can serve as models or targets for planning and executing an ecological restoration – or rehabilitation – project (SER 2004; Clewell & Aronson 2007). When no such site or system exists, it is necessary to construct a reference from available information and knowledge about what did exist in the past, within the limits of a well-defined so-called *normal functioning* (van Andel *et al*. 1987) and *historical range of variability* (Higgs 2003).

Considerations of natural dynamics and environmental changes imply that reference systems, which we know or construct from past or from present undisturbed areas relevant to our site, can serve as models to orient and inspire us, rather than as a strict objective to be literally reached. The reference serves the role of a 'guiding vision' but also provides a multidimensional yardstick or benchmark, to be used for the comparison of what is happening to the ecosystem undergoing restoration with respect to our prespecified goals for it. It is not a romantic or naive notion of trying to return somehow to the past, something we know is not possible. Inspired by the knowledge of the process of evolution of species – an ongoing process that does not start from scratch, *de novo*, but inevitably has historical roots – a restoration scientist should gather as much information as possible to understand the historical development and, where appropriate, the human transformation of the ecosystems and landscapes to be restored.

For purposes of nature conservation and restoration, in many parts of the world there are often adequate, appropriate and well-documented historical references available, sometimes even *in situ*, dating for example from the late nineteenth or early twentieth century. Often, however, for example within the vast Euro-Mediterranean region, with its intricate tapestries of seminatural and managed ecosystems, the choice or construction of a reference model is quite complicated, and polemical, involving much discussion and negotiation. We are of the opinion, however, that it is a worthwhile endeavour (see Clewell & Aronson 2007).

### 2.2.5 Zooming in on ecology

After having emphasized the need for inter- and transdisciplinarity, we now zoom in on key ecological topics. In the *SER Primer on Ecological Restoration* (SER 2004), and in Clewell and Aronson (2007), nine *attributes of restored ecosystems* are proposed for consideration. These are (1) a characteristic assemblage of the species that occur in the reference ecosystem, (2) the primary presence of indigenous species, (3) the representation of all known functional groups necessary for the continued development and/or stability of the system, (4) the appropriate physical environment to sustain reproducing species populations, (5) the normal functioning, and the absence of signs of dysfunctioning of the system, (6) the suitable integration into a larger ecological matrix or landscape, (7) the elimination or reduction of threats to the health from the surrounding landscape, (8) a sufficient resilience to endure the normal periodic stress events and (9) an ability to be self-sustaining to the same degree as its reference ecosystem. Clewell and Aronson (2007), exploring these 'desirable attributes', consider the formulation of 'standards' a useful 'broad-brush' approach, in each case to be judged in comparison to a reference system. Though they recognize that it may be impossible to meet all these criteria, they consider such standards indispensable for the evaluation of monitoring results during the trajectory towards any goal.

We recall that **ecosystem structure and functioning** are core issues in any ecological restoration project, also in cases of a focus on the reintegration of a landscape or the rescue or reintroduction of a species population. At the same time, we recognize throughout this book that understanding of ecosystem processes requires knowledge of landscape ecology, community ecology and population ecology and genetics. Therefore, each of these subdisciplines of ecology is discussed in Part 2 of this book. Here, we introduce a number of key topics in restoration ecology related to landscapes, ecosystems and biodiversity.

## 2.3 LANDSCAPES

As mentioned, the focus of ecological restoration in this book is on entire ecosystems, as per the *SER Primer's* definition (SER 2004). In Part 3 of this book, the reader will find no less than 11 applications of this philosophy. However, as we will see in several of these chapters (especially 16–19, dealing with wetlands), and still more explicitly in each of the next three chapters (3–5), a 'landscape perspective' on the restoration of ecosystems is essential. Ecosystems can be defined individually, but they do not function independently of their biotic and abiotic surroundings. Their spatial and ecological relationships at the landscape scale matter a great deal to what happens within them (Thompson 2011). Indeed, 'natural' or 'seminatural' ecosystems are part of a landscape matrix with several interacting systems, including production systems or overexploited systems, as per Figure 1.3.

In Chapter 5, on landscape ecology, the authors propose a new definition of 'landscape', as the initially clear definition, as given by physical geographers, has steadily become blurred after ecologists adopted the term 'landscape ecology' for their studies on the spatial dynamics of individual plant and animals species. Here, we would like to clarify the different positions in the scientific literature. From a human perspective, a **landscape** is commonly defined as a geographical area that can be mapped and interpreted from aerial photographs, forming a mosaic of interacting systems that may include 'natural' ecosystems, agro-ecosystems, villages, and industrial areas (cf. Turner *et al.* 2001). This notion of landscape has ecological, historical, economic and other human dimensions; spatial patterns and transport of matter and organisms are also important aspects. Ecologists studying **metapopulations**, at a landscape or regional scale, have adopted the term 'landscape ecology'. In this approach, a landscape is defined from the perspective of varying kinds of organisms (Wiens 1976), all of which move within and among habitat patches characterized by some degree of connectivity. As an analogy, the term 'landscape genetics' was recently coined to describe a study that aims at mapping how genes flow at the landscape scale (Manel *et al.* 2003). These landscape-scale ecological and genetic studies might, after all, have been better named 'spatial ecology and genetics', to avoid confusion with the geographical approach to 'landscape ecology'.

There is still more to be clarified. The **habitat** of a species may become fragmented in a landscape, resulting in habitat patches with local populations. The logic term **habitat fragmentation** (used in Chapter 7) has then also confusingly been termed 'landscape fragmentation', in Chapter 5 defined by the extent of habitat destruction. Once the reader understands how to interpret the existing literature, including the terminology used, we will have completed this guided tour of fundamental – and hopefully unifying – concepts.

## 2.4 ECOSYSTEMS

In view of the central task of ecological restoration to restore degraded ecosystems, we must reflect further upon the notions of **disturbance** and **stability** introduced already in Chapter 1. Note that these topics are applicable not only to ecosystems (e.g. in terms of nutrient cycling, hydrology etc.) but also to biotic communities (e.g. trophic interactions), populations (e.g. genetic equilibria) and individuals (e.g. health). Also, we need to introduce the reader to notions such as ecosystem functioning, ecosystem services and the functional role of biodiversity.

### 2.4.1 Disturbance and disturbance factors

The midcontinent population of the lesser snow goose (*Chen caerulescens*), which breeds in the eastern and central Canadian Arctic and sub-Arctic, and winters in the southern United States and northern Mexico, was relatively stable from 1950 to 1970, but it increased fourfold until a peak in 1998 (Abraham *et al*. 2005). This increase was largely because of increased survival in the winter areas in response to an agricultural food subsidy. Due to an expanding and increasingly intensive agriculture, they have adapted their migration pathways and largely graze on food crop residues, rice and wheat, and waste corn in particular (Jefferies & Rockwell 2002). Also, well-meaning nature–conservation managers established an increasing number of wildlife refuges in the winter areas and along the flyway, sometimes alongside agricultural fields. The inadvertent result of these coinciding changes in agriculture and conservation was that with ever greater densities, the geese over-exploited the tundra vegetation of their breeding ground, for example the coastal Hudson Bay salt marshes, which led in turn to irreversible degeneration of this formerly highly stable ecosystem to an **alternative stable state** of exposed sediment. The present pattern of vegetation loss is likely to continue in the foreseeable future (Abraham *et al*. 2005). This is an example of an ecological disturbance, defined as a long-term disordering of a constant or steady state, due to an external event or phenomenon, to which a given system is not capable of responding through its inherent resistance or resilience; the terms are explained below. We call the artificially inflated geese population a **disturbance factor**, and the resulting effect on the salt marsh a **disturbance** (see Figure 1.2). In the case of a serious disturbance of this sort, an ecosystem is often no longer stable in its previous state or condition, but moves instead to an alternative steady state (see also Chapters 6 and 21).

Please note that in this book we avoid the term *perturbation*, which originates from physics where it has a very precise meaning (indicating a small vibration), and is not taken to imply an interruption or disruption in a normal process. By contrast, in ecology, the term is often used either to denote a trigger or cause of severe disturbance or else as a close synonym of disturbance. There is no consistency in the way the two terms are used in the scientific ecology literature, and we do not concur with the two sole efforts at elucidation. To wit, we do not follow the terminology of Rykiel (1985) who proposed to use 'disturbance' to refer to causes, and 'perturbations' to refer to effects. Nor we do not agree with White and Jentsch (2001), who proposed to measure disturbance in absolute terms, for instance by the reduction of biomass of a mown grassland. To wit, the notion of 'disturbance' takes on meaning only once a reference system or state has been defined. In the case of the Arctic tundra, mentioned in this chapter, the reference was the ecosystem state before the disturbance was induced, when grazing by geese was still part of the 'normal functioning' of the vegetation.

To avoid confusion, researchers and authors often distinguish between kind, intensity, frequency and scale of disturbance. The kind of disturbance depends on the environmental factor concerned, whether living or nonliving, and whether human or nonhuman. The degree or intensity of a disturbance is determined by the difference between the new conditions and the previous steady state (or reference) conditions. Frequency is also important because of different effects from isolated, recurrent and continuous disturbing events; they can be irregular or regular and of differing durations. Finally, the scale or extent of disturbance refers to different spatial and temporal patterns, and to different levels of ecological organization: ecosystem, community, population or individual.

### 2.4.2 Stability and alternative stable states

What does the notion of stability – in other words, a long-standing steady state – imply? Since there are dozens of definitions of stability in the ecological literature, and definitions of resilience and resistance sometimes overlap, we adopt the most useful ones for our purposes when we discuss stability, and resistance and resilience (see Figure 1.2).

• **Stability** is the capacity of a system to return to a starting state following a significant change in its environmental conditions as the result of one or more 'disturbance factors'. If the initial or 'starting state' has persisted for a relatively long time, we can refer to it as a 'steady state', and it is kept within boundaries by the system's **resistance** and **resilience**. In ecological systems, a steady state is considered to be a dynamic equilibrium, not truly static or immobile.

• **Resistance** is a characteristic of systems that show relatively little response to a disturbance factor in terms of their structural and functional attributes.

• **Resilience** is a characteristic of systems that can be altered relatively easily by a disturbance factor but then regain their former structural and functional attributes in a relatively short time. The length of time a system requires to return to a former steady state is inversely related to its resilience; the faster the system returns to State A of Figure 1.2, the more resilient it is.

The choice of parameters to measure ecosystem stability is of utmost importance. Seeking to implement the aforementioned definitions of resilience and resistance, Mitchell *et al.* (2000) combined measurements of species attributes with environmental variables, which can represent attributes of either ecosystem structure or functionality. Their multivariate modelling approach, first applied to the conservation management of lowland heaths in Dorset, United Kingdom, helps assess why some ecosystems are more resilient than others.

As mentioned, a disturbed ecosystem is sometimes described as having crossed over **thresholds** or even **thresholds of irreversibility**, indicating that changes or switches have occurred that are severe and difficult to reverse without more or less important human intervention. In Chapter 20 the authors refer to this concept of 'thresholds' as a tool for determining the degree of ecosystem resilience and apply it to evaluate the effects of invading alien species. Once a threshold is passed, the system is considered disturbed. Similarly, a degraded ecosystem itself may remain in the disturbed state, that is, the alternative steady state can also be resilient through internal feedback that constrains restoration (Suding *et al.* 2004; cf. Folke *et al.* 2004). Very often removing a disturbance factor will not result in recovery of components of the pre-existing ecosystem that have been 'lost'. For example, just rewetting a drained wetland will not be sufficient to insure return of the 'original' or pre-existing species of that ecosystem to that site (Chapter 16). Similarly, reduction of nutrient loading in turbid, eutrophied shallow lakes rarely leads to a satisfactory recovery of a condition of clear water, indicative of a restored lake, even if the nutrient level is considerably reduced (Chapter 18). The discrepancy between the route to recovery or restoration and the initial route to degradation is known as **hysteresis**. Current knowledge on alternative stable states, or 'catastrophic shifts', can be helpful to understand and explain both disturbance and restoration processes, and to develop early warning signals for so-called *critical transitions* – also known in the popular literature as 'tipping points' – both in ecosystems and in human societies (Scheffer *et al.* 2009).

### 2.4.3 Ecosystem health and stress, and landscape integrity

We now consider three useful, but confusing, metaphors, often used to indicate the state of an ecosystem or a landscape as if these systems are a super-organism, which is of course not the case. Although it seems only a small step to elaborate the notion of ecosystem stability towards defining terms such as 'ecosystem health' and 'landscape integrity', these terms may be associated with an improper interpretation of holism. Similarly, the term 'ecosystem stress' is sometimes used metaphorically to describe the state of an ecosystem, as if the physiological state of an ecosystem could be compared to that of an individual organism. As scientists, we may have reservations about metaphors and analogies, but we are obliged to work with useful terms such as *ecosystem health*, *stress* and *integrity*, as they can help in communication and consensus building wherever ecological and socio-economic valuation systems meet (Aronson *et al.* unpubl. MS).

**Ecosystem health** has been described as 'the state or condition of an ecosystem in which its dynamic attributes are expressed within normal ranges of activity relative to its ecological state of development' (SER 2004). It can be ecologically evaluated in terms of the state of ecosystem functioning at a given time (Winterhalder *et al.* 2004), but socio-economic criteria should also be taken into account (Rapport *et al.* 1998). Rivers, for example, are not just ecosystems, but can also be considered as sources of clean water for drinking and washing, for industrial and agricultural purposes, as conduits for pollutants, and as places for recreation and aesthetic pleasure. **Ecosystem stress** then indicates the state of an unhealthy ecosystem, outside the optimal environmental range, which can be caused by disturbance factors such as **acidification** or **eutrophication** of a soil or a water body, or **climate change** in the atmosphere. Note that an ecosystem under stress may reveal to be resilient, resistant, or unstable (see Section 2.4.2). **Landscape integrity** can be indicated the way McIntyre and Hobbs (1999) describe *intact landscapes* as a **reference system** (see also Chapter 5). The degree of human intervention associated with intact landscapes can be extremely low, particularly in reserves managed for conservation, such as in Antarctica, but they can also be intensively managed, such as traditional agricultural landscapes in Europe. Disturbance factors may result in the fragmentation of habitats of particular species, among other disturbances at the landscape scale.

### 2.4.4 Ecosystem functioning

Before we move on to introduce the concept of 'biodiversity', we must touch upon the relationship between ecosystem stability and the species richness of a biotic community (see e.g. McCann 2000). Ecosystem stability is often measured in terms of **ecological functions**, or functioning or functionality, which includes rate of primary production, rate of decomposition or rate of nutrient cycling. Since the early twentieth century, the application of artificial fertilizers to species-rich seminatural grasslands, in order to increase ecosystem productivity, has caused a large decline in species richness. This in turn has triggered a strong and sustained investment in research on how productivity and species richness are related. Among efforts to explain the relationship between *species richness* and *ecosystem productivity*, we can distinguish two approaches, differing in perspectives on what is the cause and what is the effect: (1) 'How does species richness depend on ecosystem productivity?', a question that is inspired by a primary interest in determinants of biodiversity (see Section 2.5.1), and (2) 'How does ecosystem productivity depend on species richness?' The latter question is at stake here, as it relates to the role of **biodiversity** as contributing to **ecosystem functioning**, for example productivity (see e.g. Naeem *et al.* 2002). Some observations are in support of the *rivet hypothesis* (the majority of species essentially contributing to ecosystem productivity), whereas others favour the *redundant-species hypothesis* (only a few keystone species contribute to the productivity of the ecosystem). For the purpose of this chapter, rather than going into the details of this debate, we prefer to illustrate the unifying concepts of **functional groups**, **keystone species** and **framework species**, which – rather than species richness – play a significant role to explain or restore ecosystem functioning.

#### Functional groups

Various ecological classifications have been assessed to escape from dealing with individual species lists and to focus on ecological species groupings, for example life forms, strategies, adaptive syndromes and guilds. Currently, the term 'functional groups' is being used to indicate a grouping of individually known species in one particular class of functions (e.g. all the nitrogen-fixing plants in a community). Functional groups have been identified by multivariate techniques (often even without an indication of which type of function is associated with the species group), and by deductive methods that are based on the *a priori* statement of the importance of particular processes or properties in the functioning of an ecosystem (e.g. $C_3$ or $C_4$ grasses, and $N_2$-fixing Fabaceae, also known as Leguminosae). In restoration projects that make use of functional groups, it is often assumed that the effects of increasing species richness on ecosystem productivity work through changes in functional diversity. Indeed, ecosystem functioning in general is probably more related to the number of, and interactions among, the functional groups present at a site than to the overall species number (J. Wright *et al.* 2009).

#### Keystone species

Since the seminal review of Paine (1980), biotic interactions have been considered as the main underlying mechanism explaining the relationship between species diversity and ecosystem stability. Removal of a weakly interacting – that is, functionally insignificant – species would yield no or slight change, and removal of keystone species may have a cascade of effects on the community composition, transmitted by a chain of strongly interacting links. A keystone species is a member of a food web that has a disproportionally large effect on community structure. A key function can be due to high abundance of a species in a food web (e.g. a prey species) relative to other species, or result from having a large impact relative to the abundance of the species itself (e.g. a top predator).

Keystone species sometimes play a role in reintroduction programmes, for example in the case of the reintroduction of European beaver (*Castor fiber*) in a river flood plain (see Chapter 8). It is often, however, difficult to know what the true keystone species were or are, if at all, in cases where degradation and transformation have gone very far, especially in species-poor communities.

### Framework species

An interesting alternative approach, increasingly applied in projects of ecological restoration in tropical forest areas (see also Chapter 9), is to introduce a subset of species – called 'framework species' – to reorient an ecosystem along a desired or targeted **successional trajectory**, aiming pragmatically for a balance between the competitive exclusion of undesired species (e.g. exotics or invasives) and the facilitation of colonization of desired species. The so-called *Framework Species Method* first developed in Queensland, Australia (Goosem & Tucker 1995), and more recently in northern Thailand (Elliott *et al.* 2003), appears to be a promising approach to restoring tropical forests. In this technique, native 'framework species' (also called **foundation species**) are selected from the reference plant community on the basis of field trials and functional traits such as fast growth, high survival in exposed areas and rapid production of a dense canopy and fleshy fruit. Their relative position in forest succession (e.g. early, intermediate or advanced) can also serve as a selection criterion (Román-Dañobeytia *et al.* 2011). The goal is to assemble a group of species that can rapidly 'capture' or occupy a restoration site and attract seed-dispersing birds and mammals, and other dispersers, that will introduce seeds of additional native plant species, thus catalysing a progression towards a diverse native forest community. At the same time, this approach helps shade out unwanted invasive exotics or rapidly expanding populations of native colonizers, or it prevents their arrival on the site. Whether or not the community will be identical or even close to that of the reference system may take decades to determine. But, given the rapidity with which tropical forests grow, there at least one can hope for significant findings in a reasonably short period, and these results will in turn be of great interest for restoration in other, slower developing ecosystems as well.

## 2.5 BIODIVERSITY

Thanks to rising concerns about the loss of species richness and genetic variation within species due to human impact, the broad topic of **biodiversity** (biological diversity) has received much attention worldwide since 1992, following the adoption of the Convention on Biological Diversity (CBD), the Rio Declaration on Environment and Development, and Agenda 21 (http://www.un.org). 'Restoring biodiversity', an expression that we will not apply in this book, is obviously a central issue in the field of restoration ecology (see Falk *et al.* 1996, 2006; J. Wright *et al.* 2009). What is of importance here is the **reinforcement** or rescue of populations (Chapter 7) and the **reintroduction** of species (Chapter 8) in specific ecosystems. While species diversity and genetic diversity – measured as numbers or some mathematical formula – are neutral concepts, it is the task of restoration ecologists to value and evaluate both types of diversity in terms of 'naturalness', rarity, or risks. From a socio-economic perspective, biodiversity can be considered as a living, evolving biotic component of the stock of (renewable, and cultivated) natural capital. From an ecological point of view, however, if a species goes extinct, it is not renewable. This implies that not only the current functional aspects of biodiversity but also its evolutionary potential must be taken into account.

In the CBD, biological diversity is described as 'the variability among living organisms within species, between species and of ecosystems'. Though this definition has been adopted worldwide, we consider the inclusion of ecosystem diversity in the definition of biodiversity as confusing, because ecosystems not only are composed of biota but also contain an abiotic component. Here, following SER (2004), we focus on concepts related to restoration of (1) *taxonomic diversity* in biotic communities, among species of plants, animals and/or microorganisms, and (2) *genetic diversity* among individuals and populations within species.

### 2.5.1 Species diversity in biotic communities

It has been empirically shown in various ecosystems that species richness often exhibits a positive relationship with ecosystem productivity, with peak species richness at intermediate productivity (or intermediate disturbance). This unimodel relationship, also known as the 'intermediate disturbance hypothesis' (Connell 1978), or 'humped-back model' (Grime 1979), may result from increasing environmental limitations to the left and increasing interspecific competition to the right of the peak. Mittelbach *et al.* (2001), however, reviewed 171 published studies that revealed positive and negative relationships between the two as well. Probably, the entire productivity gradient is not always covered by the data. Nevertheless, this approach, which implies that species richness can be regulated by ecosystem management, has steadily gained attention in nature conservation (e.g. Bongers *et al.* 2009, and references therein). Application of concepts such as the 'species pool' and 'assembly rules' to ecological restoration help to identify the potential species richness in a region, and to predict the composition and interaction webs of a 'target community' following restoration.

### 2.5.2 Assembly from a species pool

In an ecosystem, biotic communities develop through a process called community assembly, in which individuals of species invade, persist or become extinct. While it is still an open question whether we can really speak of **assembly rules** as a set of principles or laws that predict the development of specific biological communities, as compared to development that is attributable to random processes, the search for applicability of assembly rules has opened up fruitful perspectives in the practice of ecological restoration (Temperton *et al.* 2004).

We adopt the approach given by Zobel (1997), who defined **species pools** at three different scales, with environmental filters in between, living or nonliving. A 'regional species pool' represents the total of species available for colonization and is defined within a large biogeographic or climatic region, extending over spatial scales many orders of magnitude larger than those of local ecosystems. The 'local species pool' is a selection from the regional pool, defined at the level of a landscape, and a further selection, the 'community species pool', is the set of species in a site to be restored. Assembly rules indicate constraints or environmental filters determining which species can occur in the community and which combinations are irrelevant.

### 2.5.3 Assembly rules

Which are the environmental filters? The term 'assembly rules' was coined by Diamond (1975), to help explain or elucidate, through experimentation and observation, the dynamic structure of stable and rapidly evolving communities based on niche-related processes. Working with this concept, Weiher and Keddy (1999) proposed to envisage two basic kinds of community patterning, with different causes: (1) environmentally mediated patterns, that is, correlations between species due to their shared or opposite responses to the physical environment, and (2) assembly rules, that is, patterns due to interactions between species, such as competition, allelopathy, facilitation, mutualism and all other biotic interactions that we know about in theory, and actually affect communities in the real world. Currently, all these processes, including the arrival of propagules, their germination and establishment, and their interactions with co-occurring species, are included in the notion of assembly rules.

Cavender-Bares *et al.* (2009) contribute to the clarification of the concept of assembly rules, by distinguishing between three perspectives on the dominant factors that influence community assembly, composition and diversity: (1) the classic perspective that communities are assembled mainly according to niche-related processes, (2) the perspective that community assembly is largely a neutral process in which species are ecologically equivalent and (3) the perspective that emphasizes the role of historical factors in dictating how communities are assembled, with a focus on speciation and dispersal rather than on local processes. Note that these different points of view are not mutually exclusive, and that it is useful to investigate the relative importance of the different hypothetical processes.

The notion of 'assembly rules' implies that the species composition of biotic communities can be explained and predicted. Independent of whether this claim is justified, the advantage of the search for assembly rules is that it helps make ecological knowledge about communities and ecosystems explicit in terms of predictions that can be tested.

### 2.5.4 Genetic diversity within species

Within-species genetic diversity is increasingly recognized as an important aspect of biodiversity (Falk *et al.* 2006; Chapter 7). It represents the adaptive potential of genotypes and the associated phenotypes to their living and nonliving environment, the basis of evolutionary processes such as adaptive differentiation and speciation. In the case of species richness, we have considered neutral and functional approaches. Likewise, genetic diversity can be valued in neutral ways (e.g. immunity to selection, or non-Darwinian evolution by random drift of genes that are not expressed in the phenotype) and by functional approaches (selection and adaptation, survival of the fittest or Darwinian evolution). In order to be applicable to species reintroduction projects pursued as part of ecological restoration (see Chapter 8), it is necessary to identify the ecological relevance of genetic variation.

A large number of techniques have been developed to quantify genetic diversity in populations of species. What is their adaptive value? Most molecular markers are neutral or nearly neutral to natural selection and patterns of variation in these markers primarily reflect the past gene flow and genetic drift. Neutral molecular variation rarely predicts quantitative genetic variation, a critical determinant of a population's *evolutionary potential* (see Chapter 21), and there is no theoretical basis for assuming that the population with the highest genetic diversity in molecular markers will be the best genetic source for restoration (McKay *et al.* 2005; Kramer & Havens 2009). We agree that a combination of molecular studies of large-scale patterns with ecological studies of **local adaptation** is required to assess the adaptive value of genetic variation. In view of application to ecological restoration programmes, McKay *et al.* (2005) raised the following two key questions (see also Chapter 7): (1) 'How will existing populations, adapted to local conditions, be affected by the introduction of novel genes and genotypes in a geographic region?' and (2) 'What is the level of genetic diversity required to ensure the long-term success of restoration projects?' Note that reinforcement or reintroduction of a local population only deals with the problem of introducing genotypes from nonlocal populations of the same species, which the reader should clearly distinguish from the introduction of alien species or even unwanted exotic invaders.

Single populations only seldom live in isolation. Increasing human impact at the level of landscapes may have resulted in **habitat fragmentation** and thus in an increasing risk of isolation of formerly interacting local populations. This has consequences for the genetic diversity and composition of the local populations, such as the risk of reduced gene flow and inbreeding effects or just the opposite (Young *et al.* 1996). As long as there is migration between two or more local populations, they can form a **metapopulation** (Hanski 1999). The body of theory and models concerning metapopulations is developing quickly, and has been applied for example to identify the minimum number of patches required for population persistence (e.g. Bascompte *et al.* 2002). In view of the fact that a metapopulation of any species may interact, in each of the local patches where it occurs, with members of metapopulations of other species, the concept of *metacommunity* has been proposed, described as 'a set of local communities that are linked by dispersal of multiple potentially interacting species' (Hanski 1999; Leibold *et al.* 2004). The development of this concept is still in a theoretical stage and difficult to apply to ecological restoration. We prefer, therefore, to confine the terminology to 'interacting populations', either in biotic communities (within ecosystems) or in metapopulations (within a landscape).

Something new in science and potentially useful in restoration ecology is the study of *phylogenetic similarity* (Cavender-Bares *et al.* 2009; Gerhold *et al.* 2011), and *phylogenetic signatures* (Verdú *et al.* 2009) at the level of biotic communities. Here the mapping and tracking efforts also include the phylogenesis of taxa, and would be applied not only to extant but also to intentionally reintroduced organisms or groups of organisms, and to the possible interactions to be expected among them. One issue of note is how and to what extent 'phylogenetically poor plant communities' respond to or 'receive' incoming species, and how well these newcomers co-exist with existing communities (Gerhold *et al.* 2011).

## 2.6 CHALLENGES

We close this chapter by referring to some of the key challenges ahead: uncertainty, contingency, chaos and unpredictability, on the one hand, and then transdisciplinary science and problem solving.

### 2.6.1 Coping with uncertainty

An inspiring essay by Hilderbrand *et al.* (2005), entitled *The Myths of Restoration Ecology*, points to a number of simplified and dogmatic 'beliefs' in ecological restoration. The goal of the authors clearly was not to discredit the field but rather to challenge practitioners and scientists to think about their unconscious assumptions and the inherent limitations of this field of action. A central point in the essay is the need to cope with uncertainty. The authors warned against selecting of restoration goals and endpoints as if there would be a specified climax, for the trajectory as if this would be repeatable in numerous locations, and for the concept of self-organization as if one could confine the restoration measures to recreating the physical variables. Based on their analysis, Hilderbrand and co-authors recommend how to move 'beyond the myths', recognizing and addressing uncertainty and contingencies. For example: (1) restoration requires periodic attention and **adaptive management**, (2) designing for resilience implies planning for surprise and focusing on a diversity of approaches, functions, and taxa, (3) goals should include multiple scientifically defensible endpoints of functional or structural equivalence and (4) invasive species should not *a priori* be ruled out, but considered with respect to their role as compared to native species.

Understanding uncertainty and unpredictability is a scientific goal in itself. In mathematics, **chaos theory** describes the behaviour of certain systems that exhibit dynamics that are highly sensitive to initial conditions. The behaviour of chaotic systems may seem to be random, but as soon as the initial conditions are known, it can be explained deterministically. Chaos theory can help explain incomprehensible observations. For example, Huisman and Weissing (1999), considering the so-called paradox of the plankton – the number of co-existing species far exceeding the number of limiting resources at equilibrium conditions – were able to explain species oscillations and chaos in mixtures of phytoplankton species by experiments and associated models in which they had started with slightly different initial conditions and in which the species had complex competitive interactions. Beninca *et al*. (2008) have experimentally shown that long-term stability is not required for the persistence of complex food webs; interestingly, however, also irregular oscillations appear to remain within certain limits. But, in their observations over 7 years, predictability remained limited to a period of 14 days only.

### 2.6.2 Science and Society

After considering the irrefutable and implacable reality of uncertainty and unpredictability in nature, we must nevertheless come back to the realm of active interventions, of doing restoration, as best we can. As noted, the holistic approach is transdisciplinary, and requires much outreach and consensus building – network building across professional, ideological and intellectual lines. To quote Cairns and Heckman (1996), 'The field of restoration ecology represents an emerging synthesis of ecological theory and concern about human impact on the natural world' (p. 167).

Let us thus round off this chapter by emphasizing that restoration ecology, which is ultimately the study of ecological restoration, can perhaps help form an alliance between science and society, in the search for a transition to **sustainability**. On the one hand, this interface between science and society opens up new opportunities, and on the other hand it implies risks. Applied sciences, or *science-in-context*, may become politicized when scientific uncertainties and societal interests are both heavily involved (Swart & van Andel 2008; Chapter 22). Looking at the 'scene' in Europe, today, exemplified by the implementation of the nature policy plan Natura 2000, Keulartz (2009) noted that the 'democratization' of nature conservation policy is fraught with problems and pitfalls and requires an adequate and professional organization of communication between the various actors. In Slovakia, for instance, many state-owned nature reserves have been given back to former private owners, which has resulted in much more limited conservation and restoration measures actually being carried out, even though there was no change in the legal protection of the sites. Similar problems could be cited in Albania, Italy, Greece and many other countries.

Kricher (2009) emphasizes that an understanding of the dynamic nature of ecology and evolution is essential to formulating environmental policies based on ethics that can help guide humanity towards a more responsible stewardship of our planet and all its ecosystems. He warns against nonscientific, 'teleological thinking', as if nature would or could have a goal, for example associated with a search for equilibrium or a 'climax'. Humans set goals, not ecosystems, and if the goals are set scientifically, they should include estimates not only of means but also of the transient nature of ecosystems, with all their uncertainties and sudden opening of opportunities. Coping with uncertainty is primarily a scientific task, to try and quantify the risks of deviation from means and equilibria, but this issue should be communicated with all the stakeholders in ecological restoration projects and programmes. Restoration ecology is a science, but restoration ecologists need to be competent and proactive when it comes to communications with society at large. Ultimately, restoration ecology, ecological restoration and the restoration of natural capital are all vital components of the emerging, transdisciplinary science and problem-solving paradigm of sustainability, wherein lies humanity's best chance for finding the way towards a sustainable and desirable future for humanity and all other life on our planet as well.